PACIFIC COOPERATIVE STUDIES UNIT
UNIVERSITY OF HAWAI`I AT MĀNOA
Dr. David C. Duffy, Unit Leader
Department of Botany
3190 Maile Way, St. John #408
Honolulu, Hawai'i 96822

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REHABILITATION OF SEASONALLY DRY 'ŌHI'A WOODLANDS AND MESIC KOA FOREST FOLLOWING THE BROOMSEDGE FIRE, HAWAII VOLCANOES NATIONAL PARK

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Rhonda Loh[1], Sierra McDaniel[1], Matthew Schultz[1], Alison Ainsworth[1], David Benitez[2], David Palumbo[1], Kimberley Smith[1], Tim Tunison[1], and Maya Vaidya[2]

[1]NPS, Hawai'i Volcanoes National Park, Resources Management Division, PO Box 52, Hawai'i National Park, HI 96718

[2]Pacific Cooperative Studies Unit, Hawai'i Volcanoes National Park, Resources Management Division, PO Box 52, Hawai'i National Park, HI 9671

File Name:

Organization Contact Information:

National Park Service, PO Box 52, Hawaii National Park, HI 96718, phone: 808-985-6098, fax: 808-985-6029

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Abstract

The Broomsedge Fire, accidentally started June 30, 2000, burned 1008 acres of native plant communities (3,800-4,100 ft elevation) in Hawaii Volcanoes National Park. The communities affected were seasonally dry ‘ōhi‘a woodland (923 ac) and mesic koa forest (85 ac). Fire is expected to dramatically reduce fire-sensitive native vegetation and stimulate fire-adapted alien grasses, thereby increasing fire potential in the burn area. Two strategies were used to re-vegetate burned communities. The strategy in former seasonally dry ‘ōhi‘a (*Metrosideros polymorpha*) woodlands (923 ac) was to establish a community of fire-tolerant native species that could co-exist with alien grasses and wildfire. The strategy in fire damaged mesic koa (*Acacia koa*) forest (85 ac) was to increase fire resistance in specific sites by rebuilding the structure of the native understory thereby reducing the risk of wildfire spreading into high priority areas. This same strategy, to increase fire resistance by establishing a thick understory beneath a strip of koa forest, was used to reduce the likelihood of fire carrying between the Park and private landowners in the nearby Volcano Golf Course Subdivision. Restoration efforts began one week after the fire and continued to 07/08/03. Approximately three thousand worker days, including 1,239 volunteer days, were spent completing the project. Thirty native plant species were established in the burn by a combination of seeding >2.7 million seeds and outplanting 18,798 individuals that were propagated in park temporary greenhouses. Along with re-vegetation, workers searched and removed aggressive alien woody species (e.g. *Myrica faya, Psidium cattleianum, Rubus argutus*) in order to prevent their establishment in the burn. Over 7,400 individuals were discovered and chemically or manually eradicated. Permanent monitoring plots were established and the vegetation measured to evaluate the success of the project. Average survivorship of outplants in the plots was >80% (all species). There was significant recruitment of four species (*Acacia koa, Bidens hawaiiensis, Dodonaea viscosa, Sophora chrysophylla*) from seed additions into the burn. By 2004, eleven re-introduced native species had reached reproductive maturity in the burn area, including four tree, three shrub, a lily, Hawaiian poppy, and two grass species. Monitoring should continue over the next few decades to evaluate long term successional outcomes as a result of the restoration project. The successful establishment of species by outplanting and artificial seeding in the Broomsedge Burn serves as a model for restoration in other fire-affected dry ‘ōhi‘a woodlands in the Park.

Introduction

The Broomsedge Fire, accidentally started June 30, 2000, burned 1008 acres of native plant communities (3,800-4,000 ft elevation) in Hawai`i Volcanoes National Park (HAVO) (Figure 1). Communities affected were seasonally dry ‘ōhi‘a woodland (923 ac) and mesic koa forest (85 ac). Fire was expected to dramatically reduce native vegetation and stimulate fire-adapted alien grasses, thereby increasing fire potential in the burn area (D`Antonio and Vitousek 1992, Tunison *et al.* 1995, D`Antonio *et al.* 2000). Two strategies were used to re-vegetate the burned communities with native plants. These were carried out in the drier ‘ōhi‘a woodlands and the mesic koa forest. A third strategy to prevent the establishment of ecologically disruptive alien woody species was also included in the rehabilitation plan.

The strategy in burned ‘ōhi‘a (*Metrosideros polymorpha*) woodlands (~900 acres) was to establish fire-tolerant native plants that could persist in competition with exotic grasses and recurrent wild fires. Since the invasion of fire-adapted invasive broomsedge (*Andropogon virginicus*) and bushy beardgrass (*Schizachyrium condensatum*) in the mid-1960's and molasses grass (*Melinis minutiflora*) in the 1970's, fire frequency has increased three-fold, and fire size 60-fold. Nearly two-thirds of the dry ‘ōhi‘a woodland system has been affected by wildfire and the remaining portions are at high risk for wildfire at HAVO. Re-introduction of formerly dominant but fire-sensitive native plants such as ‘ōhi‘a, and pukeawe was considered counterproductive given the widespread abundance of alien grasses that increased fuel loads and thereby the inevitability of future fires (D`Antonio and Vitousek 1992). Instead, the intent was to create modified native communities that were able to self-perpetuate, accepting that alien grasses and wild fire

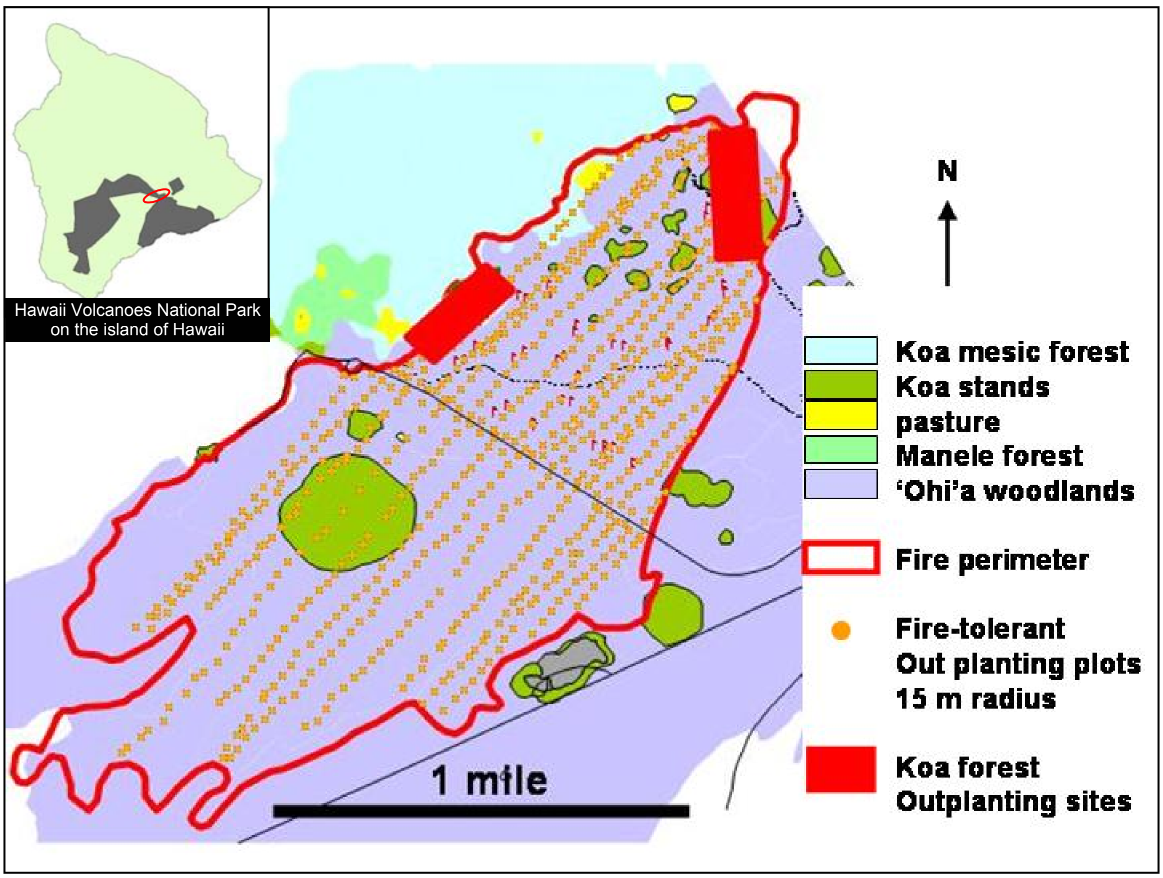

Figure 1. Distribution of planting plots in the different native plant communities affected by the Broomsedge Burn

remained important ecosystem components. Between 1993 and 2001, seven research burns and laboratory experiments were conducted to test native species capacity to survive and colonize after fire. Fourteen native plant species, among them māmane (*Sophora chrysophylla),* a tree, ‘a‘ali‘i *(Dodonaea viscosa*), a shrub, and ‘iliahi (*Santalum ellipticum* var. *paniculatum*), a tree, were identified as fire-tolerant (Loh *et al.* unpubl.). Additional species were identified based on their ability to survive or colonize after natural fire. Many of these species were formerly abundant in the seasonally dry ‘ōhi‘a lowland ecosystem, but were depleted by historic browsing by feral goats. Park boundaries were fenced from sea level to 6,600 ft elevation and goats removed in the 1970`s. However, natural recovery of species was poor because the native seed bank has been depleted, and seedlings that did establish had a difficult time growing in competition with alien grasses (Hughes and Vitousek 1993, D`Antonio *et al*.1998). Restoration in the Broomsedge Fire area focused on establishing 14 of these fire-tolerant native species by a combination of seeding and outplanting of nursery-propagated individuals in the burn (Table 1). The intent was to establish species rapidly before exotic grasses fully recover (estimate ~ 3 yrs). Koa (*Acacia koa*), a species uncommon in dry ‘ōhi‘a woodlands, was included as a restoration species because of its prolific ability to regenerate from root suckers in response to disturbance, and its natural occurrence in the area prior to the burn.

The strategy in mesic koa forest was to create a vegetated fuel barrier that would reduce the likelihood of fire penetrating Kipuka Puaulu Special Ecological Area (Figure 1). Kipuka Puaulu is a rare mesic soapberry-koa-‘ōhi‘a community that contains a number of endangered and rare plant species. Surrounding the Kipuka is degraded koa forest that is vulnerable to wild fire. These are koa forests that have regenerated in abandoned pastures following the removal of cattle in the 1950`s. Koa came up vigorously and reformed a forest canopy, but very little regeneration of the sub-canopy and understory vegetation occurred. The result was an over-simplified koa community composed of koa and alien grasses (*Ehrharta stipoides, Pennisetum clandestinum)* that was more vulnerable to wild fire. During the Broomsedge Burn, fire carried by meadow ricegrass (*Ehrharta stipoides*) burned 85 ac of koa forest and came within 50 m of Kipuka Puaulu Special Ecological Area before changing weather conditions and control lines constructed by fire fighters stopped the fire. Restoration efforts were focused on restoring the native understory that existed prior to ungulates in degraded koa forest that lie immediately adjacent to Kipuka Puaulu. Re-establishing native understory plants would create a shadier, moister environment that would suppress alien grasses and possibly reduce the risk of fire being carried in this environment and penetrating Kipuka Puaulu. The koa canopy that was damaged by fire was expected to recover naturally through vigorous production of root suckers (Parman 1976, Hauss unpubl.). Seeds and cutting of nineteen native plant species were collected and individuals propagated in park greenhouses and subsequently planted beneath the koa (Table 1).

The same strategy, to increase fire resistance by establishing a thick understory beneath a strip of koa forest, was used to reduce the likelihood of fire carrying between the Park and private landowners in the nearby Volcano Golf Course Subdivision. This is an area where young koa stands were beginning to naturally establish in ‘ōhi‘a woodlands (Figure 1). Eleven native species were propagated and subsequently planted beneath a 400 m strip of naturally recovering koa (Table 1). A subset of species was also established by artificial seeding of the area. Additional koa was planted in sections of the strip where the density of trees was sparse in order to establish a continuous strip of koa forest.

In addition to native plant introductions, ecologically disruptive alien plant species were prevented from establishing in the burn area. Faya tree (*Myrica faya*), Himalayan blackberry (*Rubus argutus*), Himalayan raspberry (*Rubus ellipticus*) and strawberry guava *(Psidium cattleianum*) are invasive weeds that have the potential to form dense stands or thickets that displace native plant communities and disrupt ecosystem processes (Smith 1985). Faya tree is a nitrogen-fixing tree that forms dense monotypic stands that excludes all native vegetation, and increases nitrogen inputs up to four-fold in invaded areas (Vitousek and Walker 1989). Strawberry guava is considered among the worst weeds in lowland Hawaiian forest (Smith 1985, Huenneke and Vitousek 1990). Individuals establish across a broad range of light environments (Loh 2004), and

Table 1. Native species for introduction into burned communities. X denotes establishment by outplanting, x denotes establishment by artificial seeding, * denotes fire-tolerant species.

Species		Pre-burn Community		
Common Name	Scientific Name	ʻŌhiʻa Woodlands	Young Koa Stands	Mesic Koa Forest
Overstory trees				
Mānele	*Sapindus saponaria*			Xx
Koa*	*Acacia koa*	Xx	Xx	
ʻŌhiʻa	*Metrosideros polymorpha*	x	x	x
Small trees				
Alani	*Melicope hawaiiensis*			x
Hōawa	*Pittosporum hosmeri*			x
Hōawa	*Pittosporum terminoides*	x		
ʻIliahi*	*Santalum paniculatum*	x	x	x
Kōlea	*Myrsine lessertiana*			x
Kōpiko	*Psychotria hawaiiensis*			x
Māmaki	*Pipturus albidus*			x
Māmane*	*Sophora chrysophylla*	Xx	Xx	Xx
Naio*	*Myoporum sandwicense*	x	x	x
Olopua	*Nestegis sandwicensis*			x
Papala	*Charpenteria obovata*			x
Pāpala kēpau	*Pisonia brunoniana*			x
Pilo	*Corposma rhyncocarpa*			x
Shrubs				
ʻaʻaliʻi *	*Dodonaea viscosa*	X	X	X
ʻĀkala	*Rubus hawaiiensis*			x
Koʻokoʻolau*	*Bidens hawaiiensis*	Xx		
Maile	*Alyxia olivaeformis*			x
Māʻohiʻohi	*Stenogyne rugosa*			x
Naʻenaʻe*	*Dubautia ciloata*	x	x	
Naupaka*	*Scaevola kilaueae*	x		
Ōhelo*	*Vaccinium reticulatum*		x	
Pukeawe	*Styphelia tameiameiae*	x	x	
ʻŪlei*	*Osteomeles anthyllidifolia*	x	x	
Herbs, ferns, and sedges				
Palapalai	*Microlepia strigosa*			x
Pua kala*	*Argemone glaucum*	x		
ʻUkiʻuki *	*Dianella sandwicense*	x		
Grasses				
ʻEmoloa*	*Eragrostis variabilis*	x	x	
*Deschampsia**	*Deschampsia australis*	x	x	

form dense monospecific stands that preclude all other vegetation. Himalayan raspberry and blackberry are major pests that form large persistent thickets or mats in seasonally dry and mesic forest (Smith 1985). These species are present in low numbers in unmanaged areas near the burn (Benitez unpubl.), and may severely limit native plant establishment if allowed to colonize in the burn. Between 2001 to 2003, work crews searched and chemically or manually removed these species in the burn.

Vegetation monitoring to evaluate recovery of the vegetation and the efficacy of the restoration efforts was conducted between August 2001 and April 2004. Included were measurements of native outplant survivorship, recruitment from seed additions, and natural recovery of the vegetation community following wildfire.

Methods

Native Plant Restoration

ʻŌhiʻa woodland

In the ʻōhiʻa woodlands area, restoration of native plants was concentrated in 693 plots that were established along 52 transects that spanned the burn (Figure 1). Plot density was ~ 4 plots/hectare). Transects started at the Mauna Loa Strip Road and extended perpendicular from the road in both directions (40, 220 degrees). Transects were labeled A through Z east (40 degree bearing) or west (220 degree bearing). Circular plots (15 m radius) were established at 50 m intervals along each transect.

Addition of 2,000-3,000 seeds/plot of ʻaʻaliʻi (a shrub) and 400-600 seeds/plot of māmane (a tree) were conducted in all of the plots (Figure 2). Fifty to a hundred seeds of koʻokoʻolau (*Bidens hawaiiensis*) (a shrub) and 200-300 seeds of koa (tree) were added to a subset of plots (Table 2). Seeds were scattered and mixed in the soil layer ($\leq$1 cm depth) by lightly turning over the soil with a rake or potato digger. The first round of seeding into plots occurred between August 2000

Figure 2. Community volunteers broadcasting native seeds in the Broomsedge Burn, Fall 2000 (photo by Loh).

and February 2001. A second round of seeding took place in 2002 and 2003. Recruitment from seed addition was monitored by counting the number of seedlings in 40 plots. Seedlings were categorized by height (<10, 10-50, 50-100, >100 cm height) and a subset of individuals (>10 cm height) was tagged to look at one year survivorship. Twenty monitoring plots were also established away from seeded plots to serve as unseeded control plots. Monitoring of seedling recruitment took place in August 2001 and June 2003.

Planting of 15 native plant species was conducted in 450 plots, including 13 fire-tolerant species (Table 2). Number of plants varied between 4 to 57 individuals per plot. Average plot density was 23 individuals per plot. A limited number of ʻōhiʻa and pukeawe were planted in the burn. These species are considered relatively fire-intolerant, but were planted because they were the major components in the pre-burn community. Fire tolerant ʻaʻaliʻi was not planted because recruitment from artificial seeding into plots was expected to be high based on previous studies (Loh *et al.* unpubl.). Planting first began in December 2000 and continued to July 2003 as individuals propagated in the nursery matured and became available for planting in the burn. There were a total of thirty outplanting sessions. Survivorship of planted individuals, their vigor, and reproductive status, was monitored in 60 plots in March 2002 (halfway through the outplanting sessions) and in April 2004 (8 mo. after the last round of outplanting).

<u>Koa-ʻōhiʻa mesic forest</u>
Restoration of native understory plants was concentrated in a 50 m x 400 m strip of burned koa forest that lay adjacent to Kipuka Puaulu SEA (Figure 1). The planting area was subdivided into 75 plots (approx 10-15 m radius) within which planting and seeding was concentrated. Eighteen native species were planted between August 2000 and May 2003 (Table 1). Total plant density in the 50 x 400 m strip was 2,520 indiv/ha. Plots were seeded with ʻaʻaliʻi (500-1500 seeds/plot), māmane (200-600 seeds/plot), and mānele (*Sapindus saponaria*) (50-150 seeds/plot). Herbicide control of meadow ricegrass with 1% Roundup in water was performed quarterly in order to minimize competition from alien grasses while planted individuals established. Survivorship of nine planted species was monitored in March 2002 or midway through the project. Monitoring of seedling recruitment from seed addition was not conducted.

<u>Young koa stands</u>
Establishment of koa and native understory plants was concentrated in a 50 m x 400 m burned strip that lay ~100m west and parallel to the Park boundary adjacent to the Volcano Golf Course Subdivision. The planting area was subdivided into 60 plots (approx 15 m radius) within which planting and seeding was concentrated. Natural recovery of fire-damaged koa stands was augmented with outplants of koa and 10 other native species between March 2002 and February 2003 (1,327 indiv/ha). Seed addition of koa (250-900 seeds/plot), māmane (1,400-2,200 seeds/plot) and ʻaʻaliʻi (4,000-10,000 seeds/plot) was conducted in all the plots. No outplant survivorship or seedling recruitment was monitored in this site.

<u>Seed collection, seed preparation, seed storage</u>
Plant material for propagation and seeding in the burn was collected from individuals and populations located closest to the burn. Material was collected from as many individuals as possible in order to maximize genetic diversity. For common species, material was collected from >100 individuals. For less common species, material was collected from at least 25 individuals. Fewer than 10 individuals were used as source material for locally rare native tree species, alani (*Meliocope radiata*) and kōlea (*Myrsine lessertiana*), that were planted in mesic koa forest.

Fruits were generally pulped and removed from the seed before broadcasting into the burn area or for plant propagation. Seeds of ʻaʻaliʻi remained in their capsules for broadcasting in the burn area. Seeds were stored in refrigerators with the exception of ʻaʻaliʻi and the legumes, which were stored at room temperature before they were husked.

Table 2. Number of outplants and seeds added to the burn by species

Common Name	Scientific Name	Total number of planted individuals	Total number of seeds added to burn
Overstory trees			
Mānele	*Sapindus saponaria*	691	5,250
Koa	*Acacia koa*	1,794	67,650
ʻŌhiʻa	*Metrosideros polymorpha*	320	
Small trees			
Alani	*Melicope hawaiiensis*	24	
Hōawa	*Pittosporum hosmeri*	75	
Hōawa	*Pittosporum terminoides*	16	
ʻIliahi	*Santalum paniculatum*	234	
Kōlea	*Myrsine lessertiana*	32	
Kōpiko	*Psychotria hawaiiensis*	58	
Māmaki	*Pipturus albidus*	946	
Māmane	*Sophora chrysophylla*	3,682	455,850
Naio	*Myoporum sandwicense*	798	
Olopua	*Nestegis sandwicensis*	10	
Papala	*Charpenteria obovata*	75	
Pāpala kēpau	*Pisonia brunoniana*	254	
Pilo	*Corposma rhyncocarpa*	1,801	
Shrubs			
ʻaʻaliʻi	*Dodonaea viscosa*		2,197,500
ʻĀkala	*Rubus hawaiiensis*	135	
Koʻokoʻolau	*Bidens hawaiiensis*	514	8,850
Maile	*Alyxia olivaeformis*	282	
Māʻohiʻohi	*Stenogyne rugosa*	247	
Naʻenaʻe	*Dubautia ciloata*	598	
Naupaka	*Scaevola kilaueae*	408	
ʻŌhelo	*Vaccinium reticulatum*	46	
Pukeawe	*Styphelia tameiameiae*	428	
ʻŪlei	*Osteomeles anthyllidifolia*	1,507	
Herbs, ferns, and sedges			
Palapalai	*Microlepia strigosa*	127	
Pua kala	*Argemone glaucum*	77	
ʻUkiʻuki	*Dianella sandwicense*	340	
Grasses			
ʻEmoloa	*Eragrostis variabilis*	1467	
	Deschampsia australis	1712	

Plant propagation was conducted in temporary park greenhouses that were constructed for the project. A 100 x 24 ft hoop house, and two 20 x 20 ft nurseries were constructed by March 2001. Twenty-four 4x12 ft benches were built and, combined with 30 benches provided by the park, were devoted to propagation of native plants for outplanting into the burn. Many of the species had not been propagated or planted in the park in recent years (e.g. *Santalum paniculatum, Charpenteria obovata, Melicope radiata*). Plant propagation techniques had to be developed or refined for the rehabilitation project. Propagation techniques and sanitation protocols developed for the project are currently being summarized in a separate technical report (S. Mcdaniel pers. comm.).

Alien Plant Control

Work crews conducted systematic sweeps on foot and removed alien woody species, Himalayan blackberry (*Rubus argutus*), Himalayan raspberry (*Rubus ellipticus)*, strawberry guava (*Psidium cattleianum*) and faya tree (*Myrica faya*) from the entire burn area. Individuals were either manually uprooted or stems were cut and Garlon 3a (10% in water) applied to stumps. Treatment was conducted annually between 2001 and 2003.

Monitoring Natural Recovery in the Burn Area

Additional monitoring to evaluate community response to fire and rehabilitation activities was conducted in permanent plots. Sixty 30 x 20 m plots were established in the burn, forty inside planting plots and twenty outside the planting plots to document vegetation changes following fire and management. Half of the plots were located in an area that had been twice burned (2x burn), once in 1977 and again during the Broomsedge Burn. The remainder was established in an area that had only burned once during the Broomsedge Burn (1x burn). Ten additional 30 x 20 m control plots were established in adjacent unburned ‘ōhi‘a woodlands. Inside the plots, vegetation cover, native seedling recruitment and tree survivorship were recorded, following modified protocols described in the Fire Monitoring Handbook (NPS 1992). Modifications to the FMH protocols included 1) reduction of plot size (20 x 30 m), and 2) trees categorized by basal diameter rather than DBH. Monitoring was conducted in summer 2001, one year after the fire, and at the end of the project in 2003. Also, recovery of native ‘ōhi‘a (n=206) and koa (n = 97) individuals were monitored along 6 transects that spanned the burn. Fire severity (char height, scorch height, percent canopy scorch) on individuals was quantified within the first month following fire, and individual survivorship (as evidenced by re-growth at the base or along the trunk and branches) was monitored one year later.

Data Analysis

Survivorship of planted species was summarized as the percent of live individuals among the total number of individuals that were monitored across all sample plots within a community type (‘ōhi‘a woodland, koa mesic forest).

Seedling recruitment from seed additions into plots was summarized as the average number of individuals (by height class, or across all height classes) per plot present in seeded and unseeded plots. Extremely low recruitment in unseeded plots prevented statistical comparison between seeded and unseeded plots. Survivorship of tagged seedlings was summarized as the percent of live individuals among the total number of individuals that were monitored across all sample plots.

To evaluate natural recovery of plants in the burn, a one-way ANOVA (SYSTAT ver. 9) was used to compare plant abundances among the 1x burn, 2x burn and unburned plots. For cover and density data, species were grouped and analyzed by life form category (exotic herb, exotic grass, exotic woody, native herb, native woody, native grass). Count data were log-transformed and cover data were arcsine transformed. Tukey’s multiple comparison tests were performed to identify significant treatment responses across sites.

Survivorship of ʻōhiʻa, and koa was summarized as the percent of live individuals (as evidenced by basal or epicormic live foliage) among the total number of individuals that were monitored along transects. Average, minimum and maximum char height, scorch height and percentage of crown scorch of individuals were summarized for each species.

Results

Native Plant Restoration Treatment

Approximately 3000 worker days (1239 volunteer days) were spent on restoration efforts in the burn. The majority of the time was devoted to collecting seeds, propagating individuals, sowing seeds and planting in the burn area. Eight hundred and twenty-eight planting plots were established, 75 in koa-ʼō hiʻa mesic forest, 693 in ʻōhiʻa woodland, and 60 in young koa stands. Nearly nineteen thousand (18,798) individuals of 30 species were planted and 2,735,100 seeds of five species were sown in the burn.

Survivorship was high among outplants despite no supplemental watering in the field. Average survivorship across all 16 species planted in fire-damaged ʻōhiʻa woodlands was 81% (Table 3). Survivorship among species ranged from a high of 96% for koa trees, to 60-85% for most other species and a low of 32% for ʻemoloa grass (*Eragrostis variabilis*) and 33% for pua kala (*Argemone glaucum*). The low survivorship of the latter two species is not unexpected since pua kala is an annual species that dies after seed production, and the ʻemoloa was difficult to relocate amid the tall alien grasses (planted individuals were not flagged). First year outplant survivorship for nine species monitored in mesic koa forest was much higher (> 90%, Figure 3) (data not shown). However, monitoring was done midway through the outplanting program and numbers of planted individuals were low for some species (e.g. < 10 indiv). More monitoring is needed to reliably determine planting success for all species planted in the koa mesic forest. No monitoring was conducted for individuals planted in young koa stands. This environment is very similar to ʻōhiʻa woodlands, and outplant survivorship is expected to be similar to survivorship in plots located in fire-affected ʻōhiʻa woodlands.

Figure 3. Native understory species planted beneath mesic koa forest May 2002. All plants are planted except for large trees. Survivorship was >90% for the nine species monitored midway through the outplanting project.

Table 3. Outplant survivorship of 16 species planted in fire-damaged ʻōhiʻa woodland of the Broomsedge Burn area. Survivorship was sampled in 60 planting plots or 13% of plots. N = number of individuals that were monitored.

Species	Scientific Name	N	% Survival	Fruit production/or root suckers
Trees				
Hōawa	*Pittosporum terminoides*	16	100	No
ʻIliahi	*Santalum paniculatum*	49	84	No
Koa	*Acacia koa*	168	96	Yes
Māmane	*Sophora chrysophylla*	311	69	Yes
Naio	*Myoporum sandwicense*	67	78	Yes*
ʻŌhiʻa	*Metrosideros polymorpha*	25	100	No
Shrubs				
ʻaʻaliʻi †	*Dodonaea viscosa*			Yes
Koʻokoʻolau	*Bidens hawaiiensis*	45	82	Yes
Na\`ena\`e	*Dubautia cilioata*	62	63	Yes*
Naupaka	*Scaevola kilaueae*	50	84	No
Pukeawe	*Styphelia tameiameiae*	46	78	No
ʻŪlei	*Osteomeles anthyllidifolia*	156	97	No
Herbs, ferns, and sedges				
Pua kala	*Argemone glaucum*	25	32	Yes*
ʻUkiʻuki	*Dianella sandwicensis*	53	94	Yes
Grasses				
	Deschampsia nubigena	155	83	Yes
ʻEmoloa	*Eragrostis variabilis*	106	33	Yes
Total	Total	1334	81	

* Fruit production was observed for these species outside the time frame of the sampling period or in unsampled plots.

† ʻAʻaliʻi was established by artificially seeding plots in the burns. Individuals were reproductively mature at the time of the last sampling.

There was significant seedling recruitment from seed additions into plots for koa, māmane, ʻaʻaliʻi, and koʻokoʻolau located in fire-affected ʻōhiʻa woodlands (Figure 4, 5 and 6). Seedling survivorship of tagged individuals > 10 cm height between August 2001 and June 2003 was 35% for māmane, 69% for koa, 98% for ʻaʻaliʻi, and 100% for koʻokoʻolau. Recruitment from seed addition into plots was observed but not quantified for mānele (*Sapindus saponaria*), māmane, and ʻaʻaliʻi in mesic koa forest, and for māmane and ʻaʻaliʻi in young koa stands.

Individuals of ʻaʻaliʻi and ten planted species were reproductively mature (seed production or root suckers) at the time of the last sampling (April 2004, Table 3). These included 4 tree species (māmane, mamaki, naio, koa by root suckers), 3 shrubs (ʻaʻaliʻi, nae nae, koʻokoʻolau), a lily (ʻukiʻuki), pua kala or the Hawaiian poppy, and 2 grasses (*Eragrostis variabilis*, *Deschampsia nubigena*). Seed production was widespread among planted koʻokoʻolau and grasses, but less common for other planted species. Planted individuals of naio (*Myoporum sandwicensis*) and māmaki (*Pipturus albidus*), small trees, were producing fruit in mesic koa plots but not in dry ʻōhiʻa woodland plots. Future monitoring will determine whether seed production will lead to successful seedling establishment in the plots.

Figure 4. Seedling recruitment of ‘a‘ali‘i from seeds placed in plots in fire-damaged ‘ōhi‘a woodland.

Figure 5. Seedling recruitment of māmane and mānele from seeds placed in plots in fire-damaged mesic koa forest.

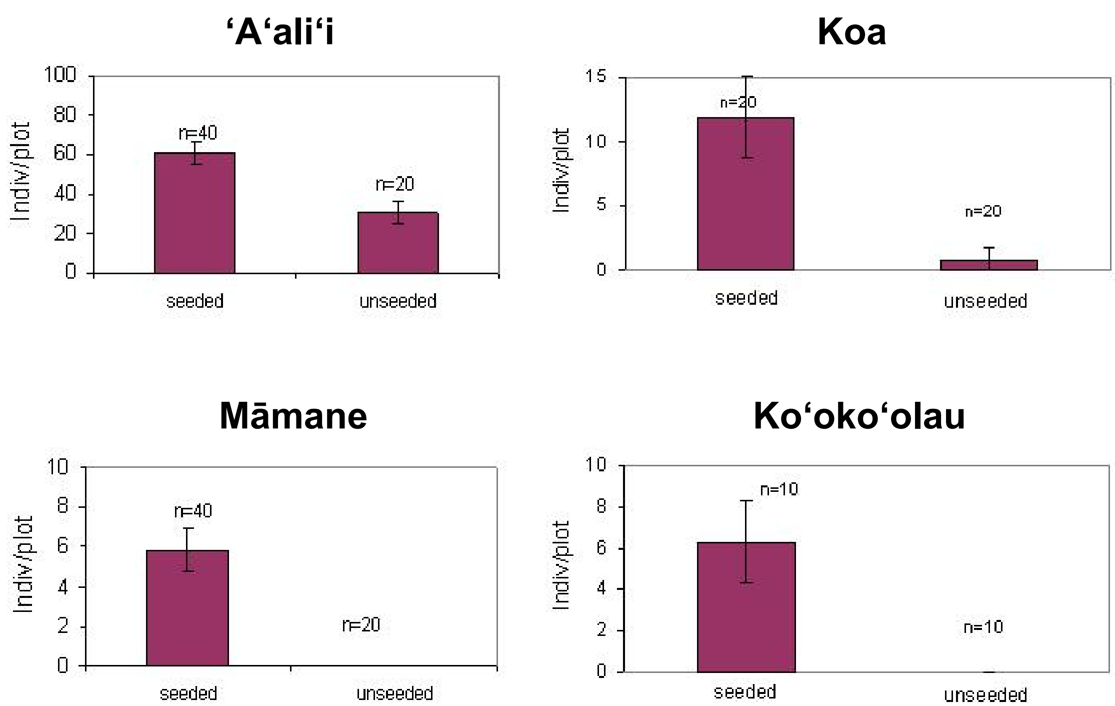

Figure 6. Seedling recruitment (all size classes) in seeded and unseeded plots located in fire-affected `ohi`a woodlands, Broomsedge Burn.

Alien Plant Treatment

Ten percent (311 worker days) of work time devoted on the projects, was spent searching and removing disruptive alien species that threatened native plant recovery in the burn. Over seven thousand (7,409) individuals of Himalayan blackberry (*Rubus argutus*), Himalayan raspberry (*Rubus ellipticus*), strawberry guava (*Psidium cattleianum*) and faya tree (*Myrica faya*) were found and removed between summer 2001 and summer 2003. Spot application of herbicide on meadow ricegrass (*Ehrharta stipoides*) was used to reduce competition to native outplants in mesic koa forest. Work crews consisted of personnel form the Division of Resources Management, contract employees through cooperative agreements, Youth Conservation Corps, and volunteer biologists.

<u>Plant Community Response to Fire</u>

Three years after fire, exotic grasses had re-established rapidly in the 1x and 2x burn areas (Figure 7). Percent cover of exotic grasses was within two-thirds of adjacent unburned areas. Both species richness and cover abundance of exotic herbaceous species were much greater in the burn than in the adjacent unburned area. In contrast native woody and herbaceous species had significantly less plant cover in the burn relative to adjacent unburned areas. Density of mature native shrubs, pukeawe and ‘a‘ali‘i, was significantly higher in the unburned (8 indiv/plot for ‘a‘ali‘i, 148 indiv/plot for pukeawe) than in the 1x burn (< 1indiv/plot for ‘a‘ali‘i and pukeawe), and 2x burn (< 1 indiv/plot for ‘a‘ali‘i, and 2 indiv/plot for pukeawe). Few fire-affected ‘a‘ali‘i and pukeawe survived the burn by resprouting at the base of fire damaged individuals. Average seedling recruitment (< 10 cm height) of ‘a‘ali‘i was 20-fold greater in the 1x burn (19 indiv/plot) and 2x burn (21 indiv/plot) than in the unburned (1.5 indiv/plot) area. There was no significant difference in small seedling recruitment (12-65 indiv/plot) of pukeawe between the 1x burn, 2x burn and unburned areas. Survival by resprouting was common for ‘ōhelo (*Vaccinium reticulatum*) in the 1x burn (62 indiv/plot) and 2x burn (9 indiv/plot).

Among the 97 fire-damaged koa trees that were monitored, greater than 95% were top killed. Char height ranged between 0.7 to 3.1 m height (average char height = 1.8 m), scorch height ranged between 2.6 to 8.0 m height (average scorch height = 5.2 m), percent crown scorch was between 66 to 100% (average % crown scorch = 90%). One year after fire, 55% of individuals

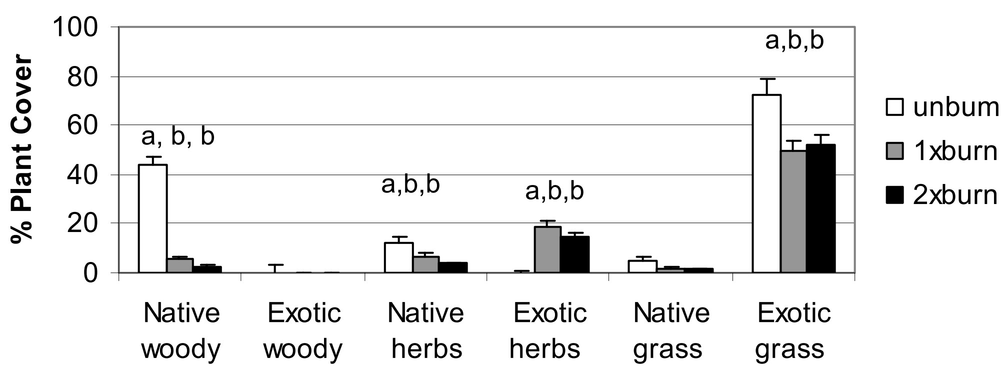

Figure 7. Average native and alien plant cover three years after the Broomsedge Burn. Reported are average absolute plant cover and standard errors in plots located in the once burn (Broomsedge Burn 2000), twice burn (Namakani Paio 1977, Broomsedge Burn 2000), and adjacent unburned areas. Treatments that share the same letter do not significantly differ at $p \leq 0.05$.

had re-sprouted (92% basal, 8% epicormic). Among fire-damaged ʻōhiʻa (n= 206), char height ranged between 0.4 to 2.8 m height (average char height = 2.3 m), scorch height ranged between 2.0 to 5.1 m height (average scorch height = 4.1 m), percent crown scorch was between 23 to 100% (average % crown scorch = 87%). One year after fire, 68% of individuals had re-sprouted (86% basal, 26% epicormic).

Discussion

Rehabilitation of the Broomsedge Burn was the first attempt by park managers to restore fire-affected dry ʻōhiʻa woodland communities at Hawai`i Volcanoes National Park. Field and laboratory experiments conducted between 1993 and 2000 had identified fire-tolerant native plant species for restoration into burn sites (Loh unpubl.). Techniques for getting species established through artificial seed banks had also been tested. The Broomsedge Burn provided managers the opportunity to apply the knowledge gained in past experiments to restoration of a large burn.

Establishment of Native Plants by Outplanting and Seeding

Native plant establishment through outplanting and seeding appeared successful based on early monitoring results. Outplant survivorship was >80% in fire-affected dry ʻōhiʻa woodlands and >90% in koa mesic forest. This is relatively high survivorship when compared to survivorship of native species planted in other parts of the park, many of which required additional watering in the field (Belfield pers. comm.). High survivorship may be partially due to more favorable moisture conditions. The burn is located on the upper elevational extent of the dry ʻōhiʻa woodlands that transitions into mesic forest. Outplant success is expected to be considerably lower in restoration projects conducted in drier parts of the ʻōhiʻa woodlands. Also, the time that elapsed between when individuals were first planted and last monitored varied between 8 mo and 3 years. Consequently, some mortality of recently planted individuals may occur in the future.

Successful seedling recruitment from artificial seeding into the burn area was evident for the five species (māmane, ʻaʻaliʻi, koa, koʻokoʻolau, mānele) where seeds were placed in plots. One year survivorship of tagged individuals (māmane, ʻaʻaliʻi, koa, koʻokoʻolau) was 35-100%, and suggests that some seedlings will continue to grow to reach reproductive maturity. Seedling survivorship in the burn area was relatively high compared to research sites in other parts of the dry ʻōhiʻa woodlands (Ainsworth unpubl., Vaidya pers. comm.), and maybe partially due to more favorable moisture conditions in the burn area. There were other species (e.g. ʻiliahi, naʻenaʻe) that might have successfully established from artificial seeding, based on observations in experimental burns and life history characteristics, but the amount of seeds available was insufficient for distributing across plots.

By 2004, individuals of ten planted species and ʻaʻaliʻi had grown to reproductive maturity in the burn area. Nine species, including ʻaʻaliʻi, showed either fruit production or regeneration by root suckers (koa) in ʻōhiʻa woodland plots. Two additional tree species (māmaki and naio) were producing fruit in mesic koa forest plots. These are positive steps toward the long-term establishment of native species in the burn area. Particularly for ʻōhiʻa woodlands, where a grass-fire cycle has established with the invasion of fire-adapted grasses, establishment of a soil seed bank of fire-tolerant species before another wild fire occurs is required to ensure the long-term persistence of re-introduced species. The assumption, based on observations in previous wildfire and research burns, is that future fires may top-kill individuals, but these fire-tolerant species will survive and proliferate through vegetative re-growth and seedling recruitment from the soil seed bank in response to fire.

Prevention of Alien Woody Species Establishment

Four of the most ecologically disruptive alien species were prevented from establishing in the burn. Over 7,000 individuals of Himalayan raspberry, Himalayan blackberry, faya tree, and strawberry guava were removed between 2001 and 2003. The intent of the search and control effort was to prevent these species from becoming established during the interim that followed the burn, when the area is largely unoccupied by plants and most vulnerable to invasion by alien species. By 2003, plant cover in the burn area had not yet recovered to adjacent unburned levels. Future monitoring will determine whether these four disruptive alien woody species will continue to invade the burn area, and whether search and control efforts need to be continued.

Natural Recovery of the Preburn Community After Fire

Natural recovery of ʻōhiʻa was relatively high compared to documented recovery in other fire-affected ʻōhiʻa woodland sites in the park (Tunison *et al.* 1995). Individuals were top-killed, but at least 2/3rds of individuals were able to regenerate from resprouting at the base of trees. ʻŌhiʻa survival through resprouting after fire varied considerably between 0% to 49% in other areas of the park, and survival appeared partially dependent on burn severity and tree size (Tunison *et al.* 1995). Occasional seedlings were observed in the burn area, but numbers were insufficient to detect in plots, and it is doubtful that seedlings will contribute much to the future recovery of ʻōhiʻa stands. This may lead to continued stand reduction with recurrent fire in the future.

Pukeawe, the dominant native shrub in the preburn community, was also heavily impacted by fire. Almost all individuals damaged by fire in plots were top-killed, and no vegetative resprouting was observed. Future recovery will depend on recruitment from the existing seed bank. Seedling occurrence was highly variable in burn plots. Seedlings grow extremely slow and have difficulty surviving in competition with exotic grasses after fire (Hughes and Vitousek 1993). Future monitoring will determine the extent to which individuals survive and grow in the presence of the exotic grasses that are recovering rapidly in the burn.

ʻAʻaliʻi and koa are two fire-tolerant species that were present in the preburn communities. Most ʻaʻaliʻi individuals were top killed and very little vegetative resprouting at the base of stems was observed. Seedlings were evident and in greater abundance in burn plots than in adjacent unburned plots. ʻAʻaliʻi shrub densities and cover increase, primarily through seedling recruitment, in response to fire (D`Antonio *et al.* 2000, Tunison *et al.* 1995). Fifty-five percent of koa re-sprouted following fire-damage in mesic koa forest. Koa is a disturbance adapted tree that regenerates through vigorous seedling recruitment and root suckers (Vogl 1969, Spatz 1973). Prolific recruitment from the natural seed bank and root suckers was evident, but not quantified, following this burn.

Conclusion

Rehabilitation of native plants in the Broomsedge Burn serves as the model for restoration in other fire-affected dry ʻōhiʻa woodlands in the Park. Over fifteen thousand acres, or two-thirds of dry ʻōhiʻa woodland have been affected by wildfire in the older section of Hawai`i Volcanoes National Park. The remaining unburned areas have been invaded by alien grasses and have the potential to burn which would result in further conversion of ʻōhiʻa woodland to alien grass savanna. Since the Broomsedge Burn, three wildfires (Kupukupu (2002), Panau Iki (2003), Kipuka Pepeiao (2004)) have affected nearly 3,000 acres of dry or transitionally dry ʻōhiʻa woodland. These include previously burned areas and large areas where fire had not historically been documented (Tunison unpubl.). Restoration of fire-tolerant native species has begun in sections of the Kupukupu burn, and plans are being developed to re-introduce fire-tolerant species in Kipuka Pepeiao.

Based on the results of this project, the following management recommendations can be made for future restoration of other fire-affected dry ʻōhiʻa woodlands in HAVO:

1. Establish seed stockpiles and plant orchards to increase plant material available for large-scale restoration. In the Broomsedge Burn area(1,008 ac), low supplies of plant material and seeds limited the pace of plant production and, in some instances, the number of individuals of a species established in the burn area. A lot of time (estimated 40% of volunteer worker days) was devoted to searching and collecting plant material for plant propagation and seeding. Collecting large quantities of seed and stockpiling them will enable managers to form a rapid response to future restoration emergencies. For species with seeds that do not remain viable in storage over years (Yoshinaga 2002), or whose seeds are extremely scarce, plant orchards should be established. Cultivation of stock material would be in outdoor nurseries or in natural areas, and would provide a steady supply of plant material for future propagation and restoration. Orchards would also protect genetic integrity of species, by preserving plant material collected from source material that may be otherwise inaccessible or vulnerable to stochastic events.

2. Continue refining and developing new techniques for plant propagation to make more effective use of limited plant material and maximize the pace and output of plant production. For the Broomsedge Burn, many of the species had not been propagated or planted in the park in recent years (e.g. *Santalum paniculatum, Charpenteria obovata, Melicope radiata*). Consequently plant propagation techniques had to be developed or refined for the rehabilitation project. Plant production increased overtime as propagators developed new techniques for cultivating plants that used limited seed supplies more effectively (e.g. germination rates were increased), employed alternatives techniques than seeds (e.g. cuttings), and increased growth and survivorship in the nursery. Only a subset of fire-tolerant species was used in the Broomsedge Burn area. These were species that had previously existed in the surrounding areas, but had been previously removed by feral goats (goats are now excluded from the park). There are other fire-tolerant species that are appropriate for reintroduction in lower elevation ʻōhiʻa woodlands (1,500-3000 ft elevation) and new fire-tolerant species are being identified in ongoing field studies. Techniques for their cultivation will need to be developed or refined for future restoration in burns.

3. Continue to monitor vegetation changes in the Broomsedge Burn area and in other related restoration experiments and projects. Among the biggest uncertainties for managers was determining the appropriate number of individuals needed to establish species in the burn. For the Broomsedge burn, the restoration strategy was to create small fire-tolerant native plant associations scattered across the burn. The assumption was that individuals would mature, reproduce, and eventually establish a soil seed bank for future proliferation with the next fire event. This jumpstart approach toward the development of an alternative fire-tolerant native plant community was perceived as the most feasible strategy for rehabilitating large expanses of dry ʻōhiʻa woodlands. The Broomsedge Burn served as a test for this model. Initial establishment of species was highly successful. However, what remains unknown are the number of species individuals, number and placement of plots, and time-required between fire intervals needed to ensure the future establishment of a fire-tolerant community. Outcomes will vary depending on site conditions, and fire frequency. Also, fire-tolerance was identified through testing seed response to small research burns and in laboratory experiments, and from observations (informal and quantitative measurements) of plant recovery from wildfires in and out of the park. The real test of fire-tolerance for these artificially created communities will be with the next wildfire. Consequently, long term monitoring in multiple restoration projects is needed to help managers determine the success of their treatments and effectively plan for future restoration projects.

   Long-term monitoring is also needed to evaluate the effectiveness of the two vegetated fuel breaks that were established in the Broomsedge Burn area. The concept of building dense vegetation to slow or stop fire spread is new to HAVO managers. Microclimate changes that enhance fire potential have been associated with the conversion of ʻōhiʻa woodlands to alien grass savannas (Friefelder *et al.* 1998). Currently, individuals planted in the vegetated strips are too small and immature to change microclimate conditions and reduce fuel loads sufficiently to slow fire spread. Monitoring will determine whether the vegetation increases and microclimate changes occur in the future. The ability of the mature fuel breaks to resist fire remains to be tested.

Finally monitoring natural recovery in burns should continue. The relatively high survivorship of ʻōhiʻa in the Broomsedge Burn area compared to previous studies underscores the need to understand the role site conditions, life history, fuel conditions, and burn severity have in determining successional outcomes. By understanding how these factors influence native species recovery, managers will be able to better predict potential recovery following fire and prioritize sites for management.

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